FINAL REPORT

Title: Impacts of multi-year drought on post-fire conifer regeneration in the Inland Northwest

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List of Abbreviations/Acronyms

MTBS: Monitoring Trends in Burn Severity
JFSP: Joint Fire Science Program
DBH: Diameter at breast height
CMD: Hargreaves' climatic moisture deficit

Keywords

wildfire; resilience; recruitment; climate change; ponderosa pine; Douglas-fir

Acknowledgements

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Abstract

Large wildfires raise questions about forest resilience in a changing climate. In this study we characterized natural post-fire regeneration in the Blue Mountains of eastern Oregon, and specifically assessed the extent to which post-fire climate drives tree seedling recruitment. Initial regeneration surveys across 184 sites indicated that post-fire seedling and sapling densities vary widely, from 0 to over 80,000 stems per hectare. Juvenile conifers were more likely to be found on sites with lower heat load (cooler, wetter topographic locations), higher overstory canopy density, and within approximately 100 m of surviving live trees (potential seed sources). These topographic and fire legacy factors were more important predictors of regeneration than mean drought indices over the first 3 years post-fire.

In order to further evaluate the effects of post-fire climate on recruitment, we destructively sampled 343 ponderosa pine and Douglas-fir seedlings and cored saplings from 8 additional sites. We then used ring counts to determine annually-resolved establishment dates. We observed fairly continuous year-to-year establishment across sites rather than episodic establishment. However, peak establishment years for both species combined had lower summer climatic moisture deficits (higher moisture availability) (Mann-Whitney U: $W = 81$, $p = 0.036$) and lower maximum June temperatures (Mann-Whitney U: $W = 81$, $p = 0.045$). Peak establishment years for Douglas-fir were also characterized by lower August climatic moisture deficit (Mann-Whitney U: $W = 90$, $p = 0.025$), and lower maximum August temperature (Mann-Whitney U: $W = 94$, $p = 0.012$). Climate did not differ between peak vs. non-peak years for ponderosa pine when analyzed alone, indicating that ponderosa pine recruitment is likely more resilient to heat and drought stress than Douglas-fir in the Blue Mountains. These findings concur with research from adjacent regions demonstrating that high temperatures and spring and summer drought conditions reduce post-fire seedling recruitment.

Despite climatic impacts, we also found seedlings generally grew to sapling size within the first 10-15 years post-fire, potentially creating fuels management challenges for forest managers. Therefore, our research suggests the Blue Mountains exemplifies a “Goldilocks problem” regarding post-fire conifer regeneration. Overall, most forested sites in the Blue Mountains currently appear resilient post-fire, and regeneration is so abundant in some areas that it may warrant fuels management. However, marginal forest sites often have no seedling recruitment one or two decades post-fire, and our findings suggest continued warming due to climate change may exacerbate recruitment limitation, potentially converting forest to non-forest.

1. Project Objectives:

1. Characterize the abundance and composition of natural conifer regeneration following stand-replacing wildfires in eastern Oregon;

Completed. The task statement of the Joint Fire Science Graduate Research Innovation Award specifies that the award is intended to "augment already planned and funded master or doctoral research to develop information and/or products useful to managers." Due to a 1-year funding delay in receiving JFSP GRIN funds, work addressing Objective 1 was supported by additional funding sources to ensure the student investigator progressed in their dissertation research. See Boag (2018), Chapter 3, for graduate thesis research findings from conifer regeneration surveys across the Blue Mountains. We refer to these findings throughout this report as they pertain to the work funded by JFSP (see Objective 2).

Boag, A. E. 2018. PhD Thesis. Climate change and wildfire: implications for forest management in the Blue Mountains of eastern Oregon. University of Colorado Boulder. Online: https://search.proquest.com/docview/2165548973/E281E1668F6C4513PQ/1?accountid=14503

2. Determine the effects of multi-year drought and site-level biotic and abiotic factors on post-fire conifer regeneration;

Completed. The student investigator used JFSP GRIN funds to complete work addressing this second objective, which enhanced the scope and applicability of their dissertation research. Specifically, in addition to characterizing the abundance and composition of post-fire natural regeneration in the Blue Mountains as part of their original dissertation research, the student investigator completed additional JFSP-funded field and lab work to destructively sample seedlings and precisely age them to determine the effects of post-fire climate on seedling establishment. This work, combined with existing dissertation research (Objective 1), tested the following hypotheses:

Ha. Fires followed by drought in post-fire years will exhibit lower seedling recruitment than fires followed by moist conditions;

Hb. Sites on south-facing slopes will exhibit more limited recruitment and lower diversity than those on north-facing aspects;

Hc. Seedling size, age and abundance will be lower further from living seed sources;

3. Understand the factors influencing post-fire management actions on private forest lands.

Hd. Private forest owners impacted by wildfire experience challenges obtaining seedling stock and funds for replanting.

Due to a 1-year funding delay in receiving the JFSP GRIN funds and the limited time before the student investigator was scheduled to finish their PhD, it was not possible to address this objective. Instead, we focused on addressing Objectives 1 and 2.

2. Background

The resilience of fire-adapted forests in the western US is being challenged by a century of fire suppression, overstory logging, and warming temperatures due to climate change. Fire suppression has largely eliminated historically frequent low and moderate severity fires, leading to greater fuel continuity and homogeneity (Parks et al., 2015). These conditions are interacting with a warming climate to cause more frequent large wildfires with large areas burning at high severity (Reilly et al. 2017).

Climate change may also increase the frequency of drought in western North America (Abatzoglou and Williams, 2016; Vose et al., 2015). Accumulating evidence suggests that interactions between drought and wildfire may convert forest ecosystems to alternate shrub or grassland states that persist for substantial periods of time (Anderson-Teixeira et al., 2013; Biggs et al., 2009; Enright et al., 2015; Lenihan et al., 2008; Scheffer et al., 2001). Post-fire forest recovery may be limited or may fail if wildfires are followed by drought, or if high-severity burn patches are too large to be naturally re-seeded by the nearest surviving trees (Harvey et al., 2015). In the last decade forest scientists observed such ecological state transitions at several sites in the southwestern US, in which historically forested areas transitioned to shrubland ecosystems following large, severe wildfires (Roccaforte et al., 2012; Savage et al., 2013). Forest conversion threatens recreation and timber-dependent rural economies, as well as the scenic beauty of western landscapes. It also impacts wildlife, increases erosion, and could reduce the ability of forests to act as carbon sinks (Allen et al., 2010).

There is increasing information on the rate and density of conifer regeneration following large and severe wildfires (e.g. Harvey et al., 2015; Kemp et al., 2016). When this project began, few studies addressed the effects of post-fire weather conditions on recruitment and empirical data were sparse (Enright et al. 2015; for exceptions see Harvey et al., 2015; Rother and Veblen, 2017). This research project builds on the existing literature by assessing post-fire recovery in the mixed conifer forests of the US Northwest's Blue Mountain ecoregion, an understudied area experiencing more frequent large wildfires (Hamilton et al., 2016). This project began as part of a Doctoral Dissertation in which the student investigator used non-destructive post-fire surveys to characterize the site-level factors affecting the abundance and composition of regeneration (Boag, 2018). This work also aimed to determine whether post-fire growing season drought conditions reduce regeneration. By using tree ring counts to annually-resolve juvenile conifer age, we evaluate the extent to which post-fire climate drives the timing of conifer establishment in the Blue Mountains. Our work addresses the growing need to understand interactions between wildfire and climate by reconstructing post-fire establishment trajectories for ponderosa pine and Douglas-fir.

If peak establishment years are closely linked to climate, this would suggest forests in the Blue Mountains may be vulnerable to ecological state transitions driven by wildfire activity and warming temperatures. Lower levels of natural regeneration may require managers to replant more extensively to maintain forest cover. Alternatively, if climate does not strongly drive natural regeneration, managers may experience the opposite problem of continuous seedling establishment and growth, which may require frequent fuels management in burned landscapes.

3. Methods

See Boag (2018) (Chapter 3) for methods used to survey post-fire regeneration at 184 sites across eight fires in the Blue Mountains in 2016 and 2017. JFSP funds supported additional fieldwork in 2018 to destructively sample and age seedlings in two of the fires originally surveyed, as well as in two additional fires. Methods and Results for this aging work (not detailed in Boag 2018) are provided below:

3.1 Study area

The Blue Mountains ecoregion (65,000 km^2) is comprised of semi-arid shrub and grassland and forested uplands in eastern Oregon, southern Washington and western Idaho. Over half the land is federally managed in national forests (Wallowa-Whitman, Umatilla, Malheur, and Ochoco) and wilderness areas. At low elevations, the forested landscape is dominated by dry mixed conifer forests comprised primarily of ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*), as well as grand fir (*Abies grandis*) in cool and moist microsites.

The Blue Mountains are topographically complex and are characterized by mountains and canyon slopes with abrupt aspect transitions. Elevation ranges from 700 to nearly 3,000-m. Most precipitation falls as snow during the winter or rain in spring and fall, while summers are hot and dry with daily temperatures at lower elevations commonly above 30°C for multiple weeks. The fire season generally runs from the end of June through the end of September, with most wildfires caused by lightning ignitions, though an increasing number are due to human ignitions (Palace, M.W. unpublished data).

3.2 Site selection

We limited potential sites to those in Douglas-fir or ponderosa pine forest types based on LANDIFRE Existing Vegetation Type (Rollins, 2009). We also limited potential sites to those < 1 km from a road for accessibility, as well as those with no post-fire logging, planting, or other post-fire management activities indicated in the Forest Service Activity Tracking System (FACTS). We then consulted with US Forest Service staff to secure permits for destructive sampling, which further limited candidate sites. We preselected candidate site coordinates in a GIS by identifying areas of moderate or high burn severity using Monitoring Trends in Burn Severity (MTBS) classes (Eidenshink et al., 2007), aiming to sample two sites per fire across four fires. We selected moderate and high burn severities to ensure sampled juveniles had established post-fire. Field sampling was designed to accurately evaluate juvenile tree age structure, not the probability of juvenile presence at each site, unlike regeneration survey methods used in Boag (2018) (Chapter 3). Therefore, in the field, researchers located the preselected point and used it as the corner of an initial 10 m x 10 m plot, and determined if it would be the left, right, and upper or lower corner of the plot using a random number list. If the center of the plot was > 100 m from a living adult tree (a potential seed source), the point was moved closer to the nearest seed source in 30 m increments until it was within 100 m. Researchers then counted all seedlings within the plot to determine if the plot contained a sufficient sample size of juveniles (a minimum of 30 individuals, based on recommendations of 20 individuals to reconstruct stand chronologies (Speer, 2010), plus an additional 10 to account for the proportion of samples that may not be able to be precisely aged (Rother and Veblen, 2017)). The plot was increased in size by 1 m in both directions along the slope contour up to a maximum of 50 m by 50 m until we obtained sufficient sample numbers.

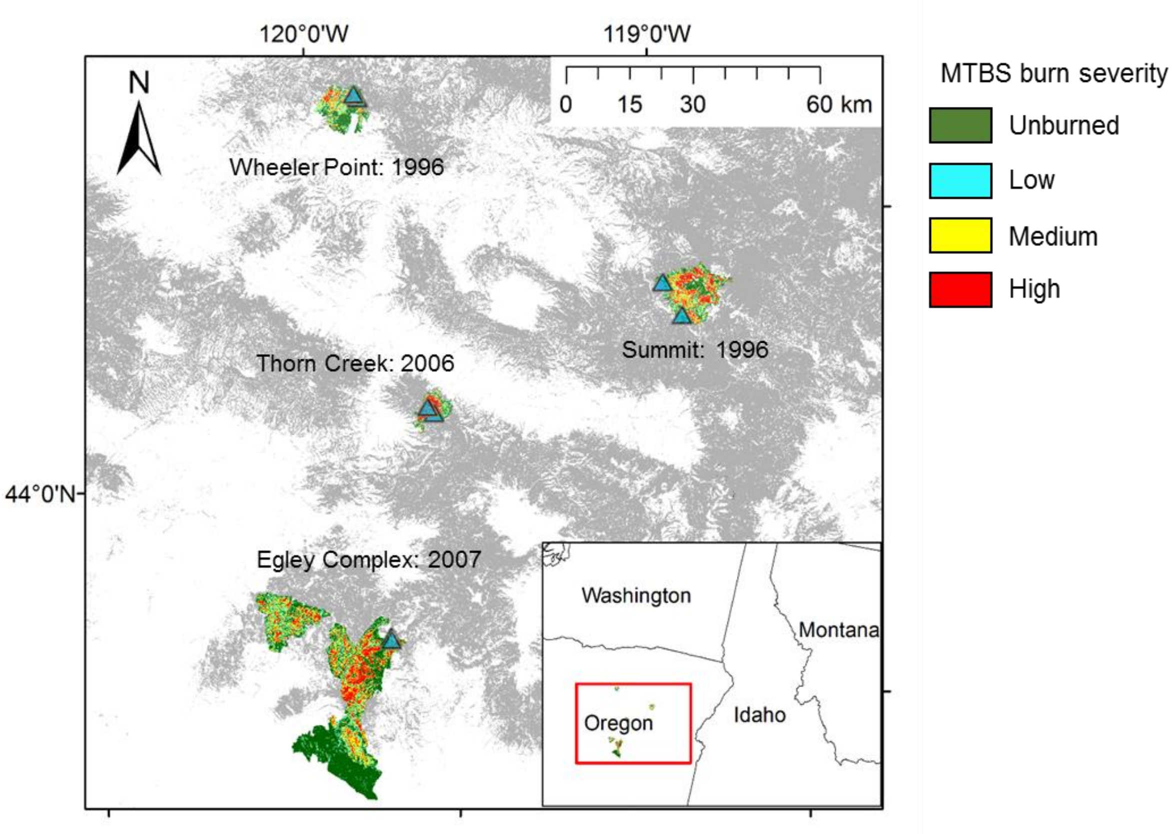

Figure 1. Site locations (triangles) of destructive seedling sampling. Fire imagery indicates burn severity (MTBS 2018).

3.3 Field sampling

In the center of each plot, researchers recorded latitude and longitude, distance to the nearest living adult conifer tree, slope, site position along the slope (lower slope, midslope, upper slope), aspect, canopy closure using a convex spherical densiometer, elevation, and took site photos.

For each juvenile tree in the plot, we recorded a tree identification number and species, and recorded whether the juvenile was located within shrub cover. We recorded the presence of any obvious herbivory or mechanical damage to the apical meristem. We then measured juvenile height and diameter at ground level (cm). We then excavated juvenile trees from the soil and cut each sample stem and roots to approximately 15 cm above and below ground-level. When juvenile trees were too large to fully excavate, we used a 5 mm increment borer to sample two cores from the stem as close to ground-level as possible, and recorded the height on the stem where each core was sampled.

3.4 Sample processing and dendrochronology

Sample stems were cut into four 2.5 cm long segments, and cores were mounted in wooden dowels with carved wells (Speer 2010). The top of each segment or core was sanded with progressively finer-grit sandpaper (80, 120, 220, 320 and 600-grit) to yield clear ring boundaries. Stem segments were then glued to cardboard mounts. Rings on each segment were counted with a 10x-40x microscope, and the precise establishment year was determined from the ring count of the last sample that still had the pith visible before it disappears into root tissue (Telewski, 1993). For samples where all segments contained visible pith, we considered this the minimum age of the seedling for use in qualitative analyses. We did not record visual marker years because they are generally considered unreliable for juvenile trees with short ring records (League and Veblen, 2006). We similarly performed ring counts on both core samples taken from each sapling, and took the largest ring count as the minimum age of the sapling for use in qualitative analyses of cohort age structure. We omitted cores and disc samples where no pith was visible in any segment from the final dataset.

3.5 Statistical analysis

As discussed above, we divided the data into two groups for analysis: precise annually-resolved ages, and minimum ages from core samples and disc samples where we did not identify the precise root-shoot boundary (i.e. pith did not disappear). First, we used precise ages from disc samples where the shoot/root boundary was confidently identified to quantitatively analyze the effects of climate on seedling establishment. Secondly, we used minimum age estimates from the disc samples where the root/shoot boundary was not confidently identified, as well as ring counts from core samples to perform qualitative assessments of establishment trajectories.

Using only the annually-resolved data, we identified peak establishment years as those with a minimum percentage (5% and 10%) of ponderosa pine or Douglas-fir establishment occurring both at each site and both sites within an individual fire (Rother and Veblen, 2017). We also identified establishment peaks for both species combined. We then acquired monthly climate data from ClimateWNA (Wang et al., 2016), and tested differences in climate variables between peak establishment and non-peak years using Mann-Whitney U tests. All data analyses were conducted in R (R Core Team, 2017). Climate variables included April-October monthly minimum, maximum, and mean temperature; total monthly precipitation; monthly Hargreaves' climatic moisture deficit (CMD) and summer (June, July and August) climatic moisture deficit (Hargreaves and Allen, 2003). CMD is a useful measure of the effects of aridity on plants, because it characterizes the moisture needed for vegetation growth from sources other than rain (e.g. soil moisture) to avoid drought impacts (Wang et al., 2016). When monthly reference evapotranspiration (E_{ref}) is less than monthly precipitation (mm), there is a moisture surplus and monthly CMD = 0. When monthly E_{ref} exceeds precipitation, CMD is the difference between E_{ref} and precipitation. Seasonal or annual CMD is therefore the sum of the monthly differences between E_{ref} and precipitation, reflecting the additive effects of drought conditions if they persist month after month.

4. Results

See Boag (2018; Chapter 3) for detailed results from regeneration surveys across 8 fires in the Blue Mountains; key findings from that work are summarized below and referenced where they pertain to findings from the destructive sampling work we conducted at 8 additional sites funded by JFSP.

4.1 Abundance and composition of natural regeneration

Objective 1: Abundance and composition of natural conifer regeneration following stand-replacing wildfires in eastern Oregon

In surveys conducted across 184 sites 15-21 years post-fire (mean = 18 years) in eastern Oregon's Blue Mountain ecoregion, we found wide variation in conifer seedling and sapling densities in the 8 fires initially surveyed (Boag 2018). Densities ranged from 0 to over 80,000 stems per hectare (median = 250); one-third of sites had zero seedlings and saplings, while one quarter of sites had densities above 2,000 juvenile stems per hectare. All sites over 25,000 stems per hectare (n = 24; 13% of sites) were "doghair" stands of serotinous lodgepole pine all located above 1,200 m (4,000 ft) elevation.

Most sites only contained one or two species (Table 1). The two most common species observed were ponderosa pine and Douglas-fir, followed by grand fir/white fir, western juniper, and western larch, lodgepole pine, and trace numbers of Engelmann spruce and subalpine fir at high elevation sites. Juvenile conifer seedling species compositions generally aligned with pre-fire forest type with the exception of sites where no regeneration was observed, which were dominated by either grasses or shrubs.

Table 1. Regeneration characteristics from 184 sites across 8 fires (Boag 2018; Chapter 3).

Years since fire*	Fire (# of sites)	Elevation range (m)	Median seedlings h^{-1}	Median saplings h^{-1}	Median total juvenile trees h^{-1} ǂ	Median species richness ǂ
21	Bull† (15)	1,456 – 1,611	3,500	10,375	14,750	4
21	Summit† (28)	1,209 – 1,503	83	83	125	1
20	Tower† (26)	1,194 – 1,583	542	208	1,208	2
20	Wheeler Point (23)	1,088 – 1,427	83	0	167	1
17	Milepost 244 (15)	863 – 1,361	0	0	0	0
16	Monument (26)	693 – 1,150	208	0	208	1
16	Hash Rock (25)	1,253 – 1,587	417	83	667	1
15	Bridge Creek (24)	871 – 1,144	0	0	0	0

*Survey year – fire year
†Serotinous response by PICO at some sites
ǂSeedlings and saplings

4.2 Effects of site-level factors and post-fire climate on recruitment

Objective 2: Effects of multi-year drought and site-level biotic and abiotic factors on post-fire conifer regeneration

4.2.1 Site-level factors

Excluding serotinous lodgepole sites, our Random Forest binary classification models indicated that the most important variables explaining juvenile conifer presence (i.e., those variables contributing to the highest mean decrease in the Gini coefficient) across the 8 fires surveyed were site heat load (a composite variable comprised of latitude, aspect and slope; McCune and Keon 2002), overstory density, and distance to live seed source (Boag 2018). Juvenile conifers were more likely to be present in sites with lower heat loads (e.g. north-facing slopes), and sites with higher overstory density (i.e. higher canopy cover of surviving adult trees). Juvenile conifers were also more likely to be present at sites closer to living seed source. These two aforementioned factors are tightly correlated with burn severity: lower burn severities result in higher overstory density and shorter distance to seed source. Saplings (2.5-12.5 cm DBH) were more likely to be found in sites at higher elevations, indicating more rapid juvenile conifer growth at higher elevations.

91% of sites < 5 m from living seed source had regeneration present, with 50% of these sites containing sapling-sized juveniles. 67% of sites 50 m or less from seed source had regeneration present. Greater than 200 m from seed source, only 27% of sites had regeneration present, with 9% of those sites containing sapling-sized juveniles. These findings suggest that recruitment occurs more slowly (seedlings are younger and smaller) in sites further from seed source.

4.2.2 Drought and post-fire climate

The site-level factors discussed above were much more important than the multi-year drought index – which we characterized using mean growing season (Apr-Sept) Standardized Precipitation Evapotranspiration Index (SPEI) of the first 3 years post-fire – in determining the presence of regeneration across the 184 sites initially surveyed (Boag 2018). However, we could not test the effects of interannual climatic variability with basic counts of regeneration, which is why we carried out the destructive sampling and aging work funded by JFSP.

For the JFSP-funded work we destructively sampled a total of 343 juvenile trees across 8 sites on four fires (Table 2; Table 3). Two-thirds of samples were ponderosa pine while the remaining one-third were Douglas-fir. There were no other tree species observed at the destructive sampling sites. 15% of juveniles sampled were cored because they were too large to be excavated.

After sample preparation and ring counts, 10% of samples were omitted from further analysis because they were either cores that missed the pith or disc samples with rot or other damage that precluded ring counts. Of the remaining samples, 32% were aged to a minimum establishment year (n = 99), with the remaining 68% aged at annual resolution (n = 210). Most of the saplings that were too large to excavate (and were therefore cored) were growing in sites on the Wheeler Point and Summit fires which both burned in 1996. Therefore, we limited our analysis of the relationship between peak establishment years and post-fire climate to the Thorn Creek (2006) (Fig. 2) and Egley Complex (2007) sites where the majority of sample ages could be annually-resolved.

By combining the annually-resolved age data with the minimum age data, establishment trajectories qualitatively indicate continuous juvenile recruitment post-fire beginning the year of or year after each fire (Fig. 3-5). While distinct establishment peaks were not immediately clear, we used annually-resolved samples to identify years in which 5% and 10% of seedlings established in both sites within the Thorn Creek and Egley Complex fires respectively (Fig. 3-5). Using a 5% filter with both species combined, peak vs. non-peak establishment years differed in summer CMD (Mann-Whitney U: W = 81, p = 0.036) and maximum June temperature (Mann-Whitney U: W = 81, p = 0.045), with lower CMD (higher moisture availability) and lower maximum June temperatures associated with peak establishment years. Several climate variables differed between peak and non-peak years for Douglas-fir alone. Peak years for Douglas-fir had lower summer CMD (Mann-Whitney U: W = 87, p = 0.039) and lower August CMD (Mann-Whitney U: W = 90, p = 0.025), as well as lower maximum August temperature (Mann-Whitney U: W = 94, p = 0.012). These findings support the hypothesis that high temperatures and spring and summer drought conditions can reduce post-fire seedling recruitment. Ponderosa pine appears more resilient to climatic variability than Douglas-fir because there were no significant differences in climate between peak vs. non-peak establishment years for ponderosa pine alone.

Finally, we used the age data to develop growth curves for juvenile ponderosa pine and Douglas-fir (Fig. 6). While the minimum age data likely underestimates true age by 1-2 years, average seedlings reach sapling size between 10 and 15 years of age.

Figure 2. Abundant post-fire ponderosa pine regeneration on the Thorn Creek fire, which burned in 2006 (photo taken in 2018).

Table 2. Site characteristics, including climate normals from ClimateWNA (Wang et al., 2016). MAP: mean annual precipitation; MAT: mean annual temperature; CMD: Hargreaves's climatic moisture deficit.

Fire	Fire Year	Site Code	Latitude	Longitude	Elevation (m)	Aspect (Degrees)	1981-2010 MAP (mm)	1981-2010 MAT (°C)	1981-2010 Climate Normal CMD
Egley Complex	2007	EG1	43.8674	-119.3202	1536	45/NE	425	5.8	594
		EG2	43.8601	-119.3161	1504	63/ENE	392	5.9	619
Summit	1996	SS3	44.7352	-118.7911	1387	190/S	476	7.1	524
		SN20	44.6763	-118.7116	1215	330/NNW	463	7	566
Thorn Creek	2006	TH22	44.3713	-119.3179	1204	35/NNE	383	8.3	622
		TH23	44.3714	-119.3303	1341	347/N	397	7.7	579
Wheeler Point	1996	WPN9	44.9575	-119.8241	1355	10/N	454	7.6	477
		WPS20	44.9673	-119.8478	1401	203/S	467	7.2	452

Table 3. Number of juvenile conifers within each plot harvested or cored for aging, and total plot stem density.

Fire	Site Code	Ponderosa pine (n)	Douglas-fir (n)	Total Juveniles (n)	Density (stems/ha)
Egley Complex	EG1	35	10	45	3,462
	EG2	9	25	34	3,091
Summit	SS3	16	14	30	632
	SN20	47	0	47	4,273
Thorn Creek	TH22	28	22	50	10,000
	TH23	32	17	49	10,889
Wheeler Point	WPN9	27	13	40	1,000
	WPS20	35	13	48	4,800
Total		229	114	343	

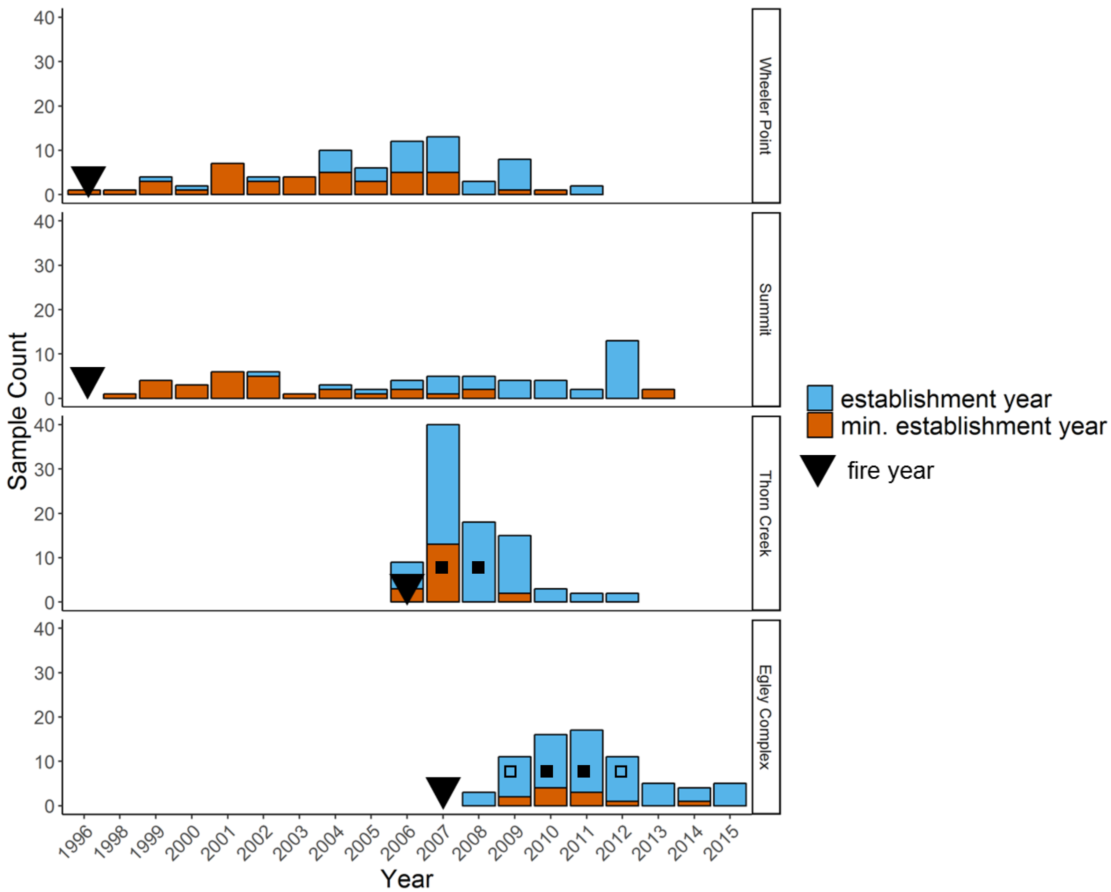

Figure 3. Minimum and annually-resolved establishment years for all juvenile trees in destructively sampled sites (n=8) across four fires in the Blue Mountains. Peak establishment years (both within sites and within fires) are indicated by squares (open = 5% filter; closed = 10% filter).

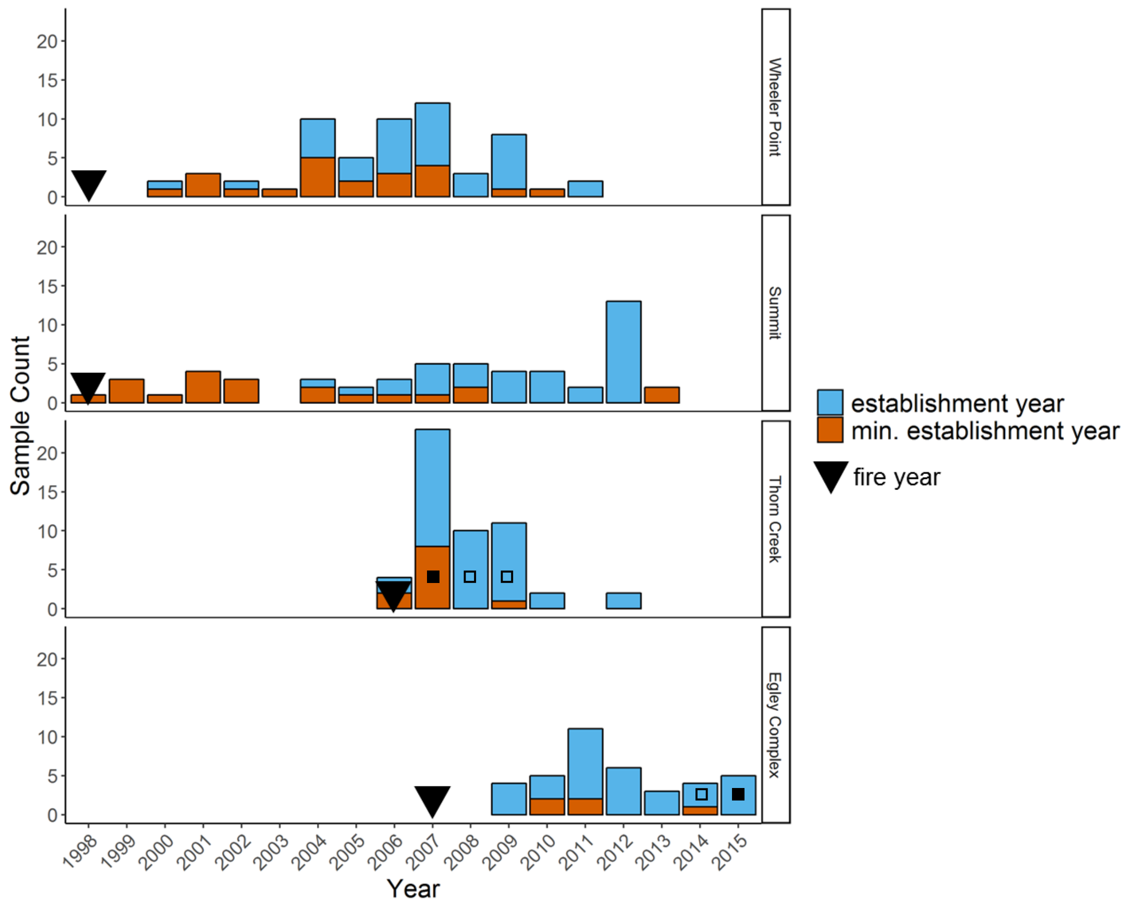

Figure 4. Minimum and annually-resolved establishment years for ponderosa pine in destructively sampled sites (n=8) across four fires in the Blue Mountains. Peak establishment years (both within sites and within fires) are indicated by squares (open = 5% filter; closed = 10% filter).

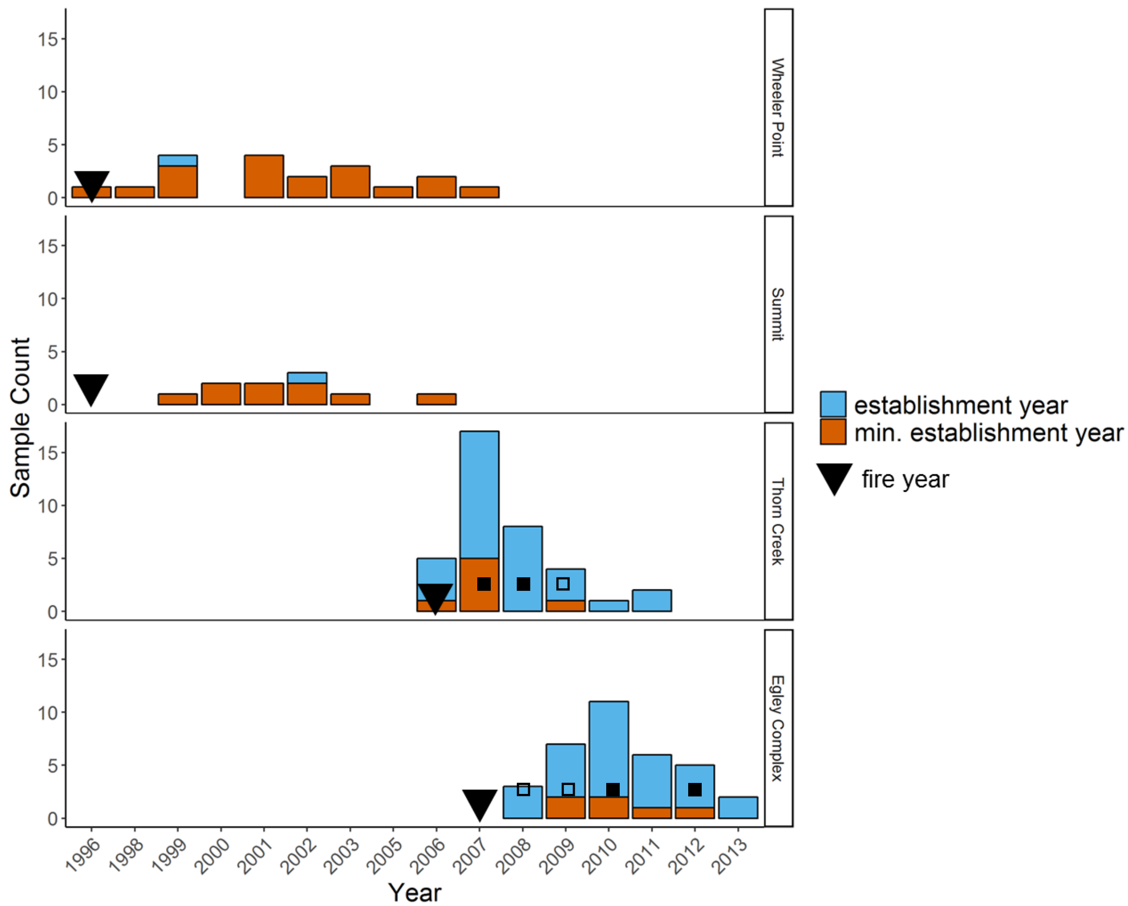

Figure 5. Minimum and annually-resolved establishment years for Douglas-fir in destructively sampled sites (n=8) across four fires in the Blue Mountains. Peak establishment years (both within sites and within fires) are indicated by squares (open = 5% filter; closed = 10% filter).

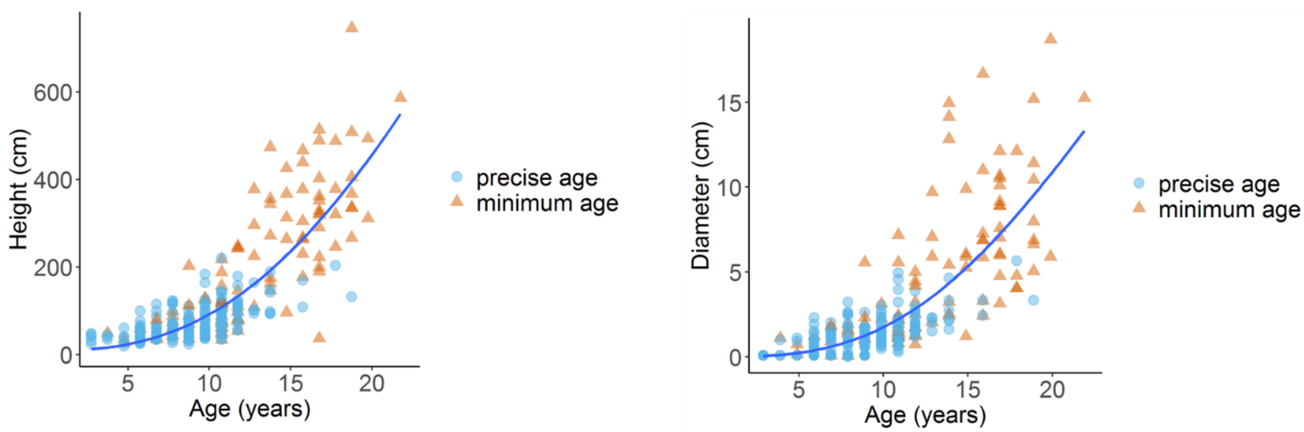

Figure 6. Height and diameter growth curves for ponderosa pine and Douglas-fir juveniles in destructively sampled sites (n=8) across four fires in the Blue Mountains. Circles indicate precisely-aged (annually-resolved) samples, while triangles indicate minimum ages. Curves are a best-fitting second order polynomial curve.

5. Discussion

This JFSP-funded project and associated research determined the main factors driving post-fire conifer regeneration in eastern Oregon's Blue Mountains. In agreement with findings from the only other widespread survey of post-fire regeneration in the region (Downing et al., 2019), seedling recruitment occurs most frequently in burned areas close to living seed source with canopy cover from surviving mature trees, and on sites with low heat load (e.g. north-facing slopes). Therefore, both local site topography and fire legacies (the distribution of surviving mature trees post-fire), which is related to burn severity, are dominant drivers of post-fire regeneration and can supersede effects of regional climate variability. In addition, at the majority of sites we surveyed juvenile conifer densities were higher than the minimum seedling stocking recommendations for planting in the region (approximately 100-300 trees per hectare) (Powell, 1999), suggesting recruitment in burned areas is currently sufficient to maintain landscape-level forest resilience. Some burned areas have incredibly high densities of seedlings and saplings (greater than 2,000 stems per ha), which may present fuels management challenges for forest managers.

To our knowledge this project is the first to reconstruct recent post-fire establishment trajectories in the Blue Mountains by destructively sampling and aging post-fire seedlings and saplings. Establishment trajectories indicate seedlings generally establish continuously each year in the first 20 years post-fire, rather than occurring episodically in some years and not at all in other years. These findings concur with results from branch whorl counts used to estimate seedling and sapling ages across 135 sites in the Blue Mountains (see Fig. 8 in Downing et al. 2019). That study also demonstrated continuous recruitment, and overall the authors found no association between establishment dates and annual climatic moisture deficit. Therefore, they concluded there is no widespread recruitment limitation related to drought conditions. However, branch whorl counts can be less reliable than bud scar counts for aging seedlings, which themselves can result in aging errors of 2 years or more compared to ring counts (Urza and Sibold, 2013).

In contrast with Downing et al. (2019), the ring count data presented here suggests that while recruitment is fairly continuous year-to-year in the Blue Mountains, peak recruitment years are associated with lower climatic moisture deficit (higher moisture availability), and lower maximum temperatures in spring and summer months. These results did not hold for ponderosa pine when analyzed alone, indicating that ponderosa pine recruitment is more resilient to heat and drought stress than Douglas-fir in the Blue Mountains. This finding aligns with recent research conducted in the Northern Rockies showing a higher mean summer temperature threshold for successful ponderosa pine recruitment (Kemp et al., 2019), and with long-standing recognition of high drought tolerance among ponderosa pine (Bates, 1923). In northern California and other sites in the Northern Rockies, studies also indicate ponderosa pine establishment is not very sensitive to post-fire weather (Harvey et al., 2016; Young et al., 2019).

These results suggest that post-fire seedling recruitment may decline as temperatures continue to increase due to climate change, and that warming will amplify the effects of existing topographic controls on recruitment. However, this research has limitations, and further work is needed to conclusively determine where forests in the Blue Mountains may be vulnerable now and in the future to climate-driven recruitment limitation.

Firstly, all the sites we surveyed and sampled were those that did not have any post-fire logging or replanting performed. Local silviculturists incorporate their understanding of where natural regeneration is likely to be sufficient when they make decisions about planting locations, and therefore our findings of fairly abundant natural regeneration may be overly optimistic (USFS staff, pers. comm.).

Additionally, in order to obtain a sufficient sample size per site to fully reconstruct establishment trajectories, we purposefully sampled locations with at least 30 seedlings. These sites likely had microclimatic, edaphic or other characteristics which promote regeneration and make these sites more productive than the one-third of sites initially surveyed which had zero regeneration (Boag 2018). To illustrate this sampling bias, Fig. 5 shows the distribution of normal climatic moisture deficit (CMD) for all sites surveyed in the Blue Mountains by Boag (2018). The destructively sampled sites discussed in this report had CMD values in the middle of this distribution. The sites surveyed by Downing et al. (2019) fell near the middle and wetter extreme of CMD values across the region's forested sites. Therefore, future research on climatic drivers of recruitment in the Blue Mountains should focus on the region's driest sites, located at low elevations and on warm aspects (normal CMD > 650). Post-fire regeneration in these sites is likely more episodic (Davis et al. 2019), and given that many sites with high heat loads had zero seedlings in our regeneration surveys, they may represent canaries in the coalmine for gradual increases in the prevalence of recruitment failure as temperatures warm.

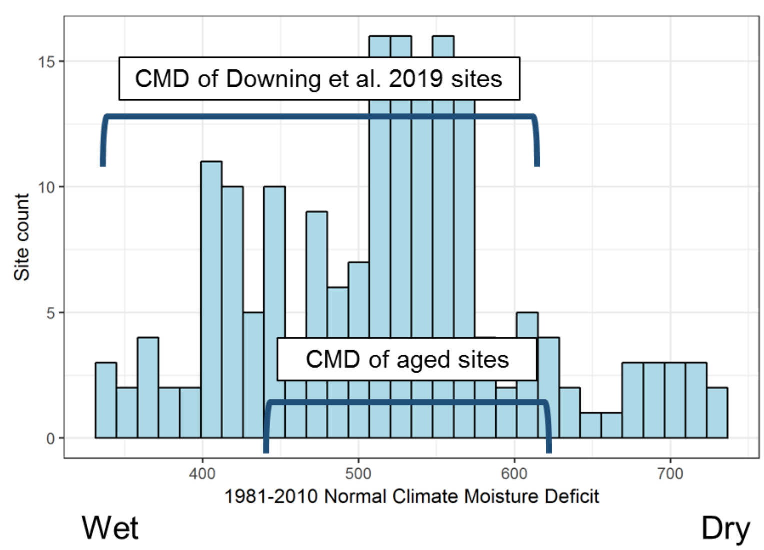

Figure 7. Climatic moisture deficit (CMD) normals (1981-2010) of all 184 post-fire sites surveyed by Boag (2018) in the Blue Mountains, with brackets indicating the CMD range of the 135 sites surveyed by Downing et al. (2019), and the CMD range of the 8 sites aged using destructive sampling described by this report.

5.1 Science delivery activities

We disseminated project findings in Spring 2019 to scientists, practitioners and managers through a conference presentation at the Northwest Scientific Association Annual Meeting in Lewiston, Idaho and a webinar hosted by the Northwest Fire Science Consortium. While we planned to co-host a tree planting workshop for landowners with Oregon State University Extension Service, late snow in eastern Oregon necessitated this workshop be replaced by a public presentation (see Appendix B for dates and number of participants). We also developed a 2-page summary guide on post-fire regeneration in the Blue Mountains, and are currently preparing two manuscripts for submission to peer-reviewed journals.

6. Conclusions

6.1 Key findings

The results of this study indicate that post-fire natural regeneration in the Blue Mountains over the last 20 years has generally been sufficient to maintain forest resilience. However, recruitment varies dramatically across sites. In burned areas with abundant surviving adult trees 100 m away or less and on north-facing slopes, hundreds or thousands of seedlings per hectare may establish within the first 10-15 years post-fire. In contrast, conifer densities in large high-severity burn patches (i.e. larger than 100-200 m in radius) with high overstory mortality, especially those on warmer sites, may be insufficient to meet local silvicultural guidelines or maintain forest ecosystem function without supplementary replanting. Some of these marginal sites may be susceptible to ecosystem state transitions to shrub or grasslands. The results of this study also suggest that as climate change causes temperatures to warm and increases the probability of growing season moisture deficits, Douglas-fir recruitment may decline in drought-prone sites. Ponderosa pine seedlings may be more resilient to warming conditions, though as warming continues they will also become vulnerable to drought stress (Kemp et al., 2019).

6.2 Implications for forest management

This research suggests forest managers may face two challenges in the Blue Mountains. Firstly, some warmer, drier sites do not contain any natural regeneration post-fire. Replanting may be needed to retain forest cover in these sites. However, with ongoing increases in wildfire size due to both landscape-scale fuel continuity and climate change, managers should consider allowing some burned areas to persist as meadow to enhance landscape-scale fuel heterogeneity. Alternatively, where planting is desirable, planting at lower densities may hedge against increasing drought stress.

At the other end of the spectrum, the trees in dry mixed-conifer forests of the Blue Mountains evolved with frequent fire, and therefore they have evolved to establish and grow quickly post-fire. This is why we observed many burned areas with incredibly high densities of natural regeneration 15-21 years post-fire. High juvenile conifer densities increase both vertical and horizontal fuel continuity, and may require fuel treatments (mechanical thinning or prescribed burning). Our growth curves indicate naturally regenerating seedlings reach sapling size 10-15 years post-fire. And, while our minimum establishment year data likely underestimates true sample ages by 1-2 years, 20 years post-fire some juvenile conifers can reach 6 m (~20 feet) in height, becoming potential ladder fuels. Experimental research suggests that saplings > 5 cm DBH can often survive prescribed burns (Battaglia et al., 2009). Therefore, in some of these highly productive sites, fuels management may be needed every 10-15 years to reduce fuel loads and control ladder fuels.

6.3 Implications for future research

This research filled a knowledge gap concerning the rate of natural conifer regeneration the Blue Mountains and its relationship to interannual climate variability. The growth curves produced from this research are also potentially useful for parameterizing regional cohort-based forest landscape models such as LANDIS-II (Scheller et al., 2007). To identify early indicators of trends towards potential ecological state transitions, future work should focus on precisely aging seedlings and saplings in sites already near climatic thresholds for specific tree species. This work will help determine when and where climatic thresholds may be crossed as the climate continues to warm.

Additional priorities for future research include investigating seldom-studied factors affecting post-fire regeneration. More research is needed to understand the timing of post-fire seed production and cone crops among surviving adult trees, the timing of locally synchronized cone crops, and how these dynamics may be impacted by the warming climate. Additionally, as more detailed soil maps (i.e. SSURGO) are completed for eastern Oregon and other incomplete areas, researchers can investigate how soil type and soil moisture dynamics effect regeneration.

On a practical note, for those conducting future destructive sampling studies to obtain annually-resolved age data, many ponderosa pine and Douglas-fir juveniles in the Blue Mountains are too large to easily excavate 10-15 years post-fire. Therefore, future work may need to target regeneration studies in fires less than 10 years old, or accept and account for the uncertainty associated with ring counts from cores taken near the root-shoot boundary.

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Appendix A: Contact Information for Key Project Personnel

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Appendix B: List of Completed/Planned Products

Completed

Conference presentation:

Boag, A. E. and J. Hartter. 2019. Growing up: Post-fire conifer establishment trajectories in eastern Oregon. Northwest Scientific Association Annual Meeting. March 26-29, Lewiston, Idaho. (15 participants)

Public Presentation (in lieu of field workshop):

Boag, Angela. April 1, 2019. Natural Regeneration Following Wildfire. Research results on post-fire seedling establishment from natural seed sources in Northeastern Oregon. La Grande, Oregon. Invited presentation (Oregon State University Extension Service). (11 participants)

Webinar:

Boag, A. E. May 1, 2019. Growing up: Post-fire conifer establishment trajectories in eastern Oregon. Northwest Fire Science Consortium. Oregon State University Extension (19 participants)

Dissertation:

Boag, A. E. December 2018. Climate change and wildfire: implications for forest management in the Blue Mountains of eastern Oregon. University of Colorado Boulder. Online: https://search.proquest.com/docview/2165548973/E281E1668F6C4513PQ/1?accountid=14503

Summary guide for managers (draft in review with collaborators):

Boag, A.E. Natural post-fire conifer regeneration in the Blue Mountains: Where, Why, and How Much?

In preparation:

Articles in peer-reviewed journals:

Boag, A.E., Hartter, J. *In preparation.* Topography and fire legacies drive variable post-fire juvenile conifer densities in eastern Oregon, USA.

Boag, A.E., Rodman, K., and Hartter, J. *In preparation.* High temperatures and low moisture availability reduce post-fire conifer recruitment in eastern Oregon, USA.

Appendix C: Metadata

This project produced a dataset containing annually-resolved and minimum establishment dates, heights and diameters for ponderosa pine and Douglas-fir seedlings and saplings. The data and metadata will be made publicly available through the Forest Service Research Data Archive (https://www.fs.usda.gov/rds/archive/) as soon as the peer-reviewed papers are published or two years after project completion. The associated metadata file has been submitted to JFSP.

Dataset Title: Eastern Oregon Douglas-fir and ponderosa pine post-fire establishment dates and dimensions

Abstract: This dataset contains ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) establishment dates, heights, and diameter data associated with JFSP Project # 17-2-01-25. The data were collected in August 2018 at 8 sites that burned at medium or high severity across 4 fires in eastern Oregon's Blue Mountain ecoregion. The Summit and Wheeler Point fires burned in 1996, while the Thorn Creek fire burned in 2006 and Egley Complex fire in 2007. All ponderosa pine and Douglas-fir juvenile conifers within variable-area plots (each encompassing a minimum of 30 individuals) were measured for height and diameter at ground level. When possible, seedlings were destructively sampled to obtain a stem sample that included the root collar. Saplings too large to be excavated were cored near ground level. Samples were sectioned and progressively sanded, then used to perform ring counts under a microscope. Annually-resolved establishment years were determined from the last section with a visible pith. All segments and core samples where pith remained visible are considered minimum establishment dates, which underestimate the establishment date by 1-2 years.